Four decades of diluting phosphorus to maintain lake quality

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Abstract

Moses Lake has been diluted for 41 years with large quantities of low nutrient Columbia River Water (CRW). Inputs of CRW during spring-summer averaged 130 x 10^6 m^3/year from 1977-1988 when spring-summer total phosphorus (TP) averaged 69 µg/L - 56% less than the predilution level (152 µg/L) in 1969-1970 with CRW input only 5 x 10^6 m^3/year. The increased input represented 1.6 volumes of the most input-affected lake area. Inputs of CRW continued through the mid-1990s at twice that rate to nearly 2.5 times since 2000 at 320 10^6 m^3/year, representing nearly 4 volumes of the affected lake area, resulting in much lower average TP (23 µg/L). Chlorophyll *a* (chl) also decreased further from 19 µg/L during the early dilution years to about 6 µg/L as TP declined. Cyanobacteria were still abundant during summer in 2017 at 43% of total biovolume, but less than during 1977-1988 (65%), and average chl was 7 µg/L, consistent with TP at 25 µg/L. Nitrogen limitation of growth rate may have lessened since the initial dilution years, with possibly less advantage to N-fixing cyanobacteria. Total and soluble N:P ratios in the lake have increased since before and after dilution began, due to decreased phosphorus in the lake as well increased inflow nitrogen.

Key words: phosphorus, dilution, lake quality, N:P ratio

Introduction

Moses Lake was hypereutrophic during the 1960s to the mid-1970s. Total P and chl concentrations averaged 152 µg/L and 58 µg/L, respectively, in Lower Parker Horn and South Lake during spring-summer 1969-1970 (Welch et al., 1992). The lake was highly degraded with average TP and chl well above the eutrophic-hypereutrophic boundaries of 100 and 30 µg/L, respectively (Nürnberg, 1996). Harmful algal blooms (HABs) of cyanobacteria, *Aphanizomenon* and *Microcystis*, occurred throughout the summer, forming unsightly surface scums, which were likely toxic (WAD)E). Although growth of algae requires nitrogen in larger quantities than phosphorus, the latter is the key nutrient that drives eutrophication of lakes worldwide and Moses Lake was not an exception (Welch, 2009).

The potential benefit of adding CRW to the lake was recognized in the early 1960s, because CRW was low in phosphorus and nitrogen and the infrastructure was in place to route CRW from the East Low Canal (ELC) through Rocky Coulee Wasteway (RCW) and Crab Creek (CC) into Parker Horn and through the South Lake outlet to irrigators downstream (Fig. 1). Transporting large quantities of CRW with a low TP concentration of about 20 µg/L would dilute the high-TP lake water and proportionately reduce the concentration of algae. There was precedent for using RCW as a feed route for irrigation water through the lake, but the quantities had been at various low rates since 1956, averaging only 5 X 10^6 m^3/year in 1969-1970 when the lake was hypereutrophic. The amount of water needed to improve lake water quality was substantial, but in-lake experiments showed that algal biomass could be reduced in proportion to CRW added (Welch et al., 1972). Thus, reduction of lake TP to the mesotrophic boundary of 30 µg/L from around 150 µg/L would require replacing most of the lake water with CRW, because phosphorus would still be entering the lake from ground water as well as recycled from bottom sediments.

Funds became available as a Clean Lakes Project from the US Environmental Protection Agency (USEPA), through the Washington Dept. of Ecology (WADOE), in cooperation with the US Bureau of Reclamation (USBR), to increase the transport of CRW through the lake with the goal of reducing spring-summer average TP to 50 µg/L and increasing transparency to 1.5 m. As a result, TP in Lower Parker Horn and the South Lake was reduced by one half during 1977-1986 when CRW input averaged 20 times the pre-project rate, and below 50 µg/L during 1986-1988 when CRW was over 30 times the pre-project rate (Welch et al., 1989; Welch et al., 1992). Total P has declined further since 2000 as CRW input averaged 60 times the pre-project rate. The lake's quality in recent years is here compared with that during the pre- and post-dilution periods.

The long-term record of CRW inflow, water quality and trophic state was possible using data from WADOE in 2001, USBR from the 1990s through 2017 and the Moses Lake Irrigation and Rehabilitation District (MLIRD) in 2017, as well as data from the University of Washington (UW) during 1969-1970 and 1977-1988. USBR data were from the lake's inlets and outlet, while WADOE and MLIRD data were from multiple sites in the lake, similar to those during 1969-1970 and 1977-1988, as well as inlets, sampled by UW personnel (Welch et al., 1989, 1992).

Site Description

Moses Lake, in eastern Washington, is a natural lake created by windblown sand dunes that historically dammed CC. The lake level was stabilized with two dams constructed in 1929 and

1963. The lake has an area of 2,790 ha with a maximum depth of 11.5 m and a mean depth of 5.6 m (Fig. 1). Most of the lake (80% of the area) is polymictic, being too shallow to permanently stratify.

The lake has two tributaries; CC that drains 80% of the watershed (5,265 km^2), which is mostly dry-land agriculture and rangeland, as well as some irrigated land (112 km^2), and Rocky Ford Creek (RFC) that is largely spring fed. The two streams contribute a similar quantity of inflow to the lake (Jones and Welch, 1990). Under normal inflows, with no CRW added, the lake would have a water residence time of about 1 yr if April–September inflow were considered a year (Welch et al., 1992).

Parker Horn (sites 5 and 7) and South Lake (site 9), between I-90 and the outlet, represent 27% of the lake's area and data from sites 7 and 9 were used to depict changes in water quality because most of the CRW inflow passes through those sections (Fig. 1). The lower part of the Rocky Ford Arm (RFA, site 8) represents another 14% of the lake's area and also received CRW with water quality similar to Parker Horn in 1977-1988. Parker Horn (5, 7), Cascade (site 8), South Lake (9) and Lower Pelican Horn (site 10) represent about half the lake's volume that is most affected by CRW input.

Water Sample Collection and Analysis

The lake was sampled twice monthly at eight sites from April through September during 1969–1970 and 1977–1988 (Bush et al., 1972; Welch et al., 1992). Water was collected at a depth of 0.5 m across a series of transects (Fig. 1). Discrete samples were also collected monthly at 0.5 m at similar sites during April-August in 2001 by the WADOE (Carroll, 2006). On the same occasions, samples were collected from RFC (13), the inflow to RFA, and from Lower CC (4), the inflow to Parker Horn that combined Upper CC (3) flow with CRW entering from the ELC (1) via RCW (2).

Water samples for soluble nutrients were filtered (0.45-µm) at the site during the 1970s-1980s. Soluble reactive P (SRP) was determined by the acid molybdate heteropoly blue method, NO_3-N by cadmium reduction and Total N as NO_3-N after persulfate digestion (EPA, 1979). Total P was determined as SRP in previously frozen samples after persulfate digestion (Strickland and Parsons, 1972). Chlorophyll a (chl) was determined by the fluorometric method, corrected for phaeophytin, through 1986 and spectrophotometrically thereafter. Due to instrument problems, values for 1984 were determined from a regression of algal biovolume on chl using past data. Methods for chl and nutrients during the 1970s-1980s are described in more detail in Welch et al. (1992).

Water samples were collected in 2017 by MLIRD personnel at a depth of 0.5 m at eight lake sites (except 8) during May-September (Fig. 1). Sampling at the upper end of RFA (not shown) was not begun until July. Inflows were sampled at two sites on CC (1, 4) and ELC (1). Sample frequency was twice in May, June and September and once in July, August and on October 2. Samples were shipped on ice to IEH Analytical Laboratories, Seattle, WA, for analysis of TP and NO_3-N. Chlorophyll-a (chl) was determined in lake samples following filtration in the IEH laboratory. Analytical procedures were according to standard methods (Eaton et al., 2005). Specific conductance (SC) was determined *in situ* with a sonde at all sites coincident with water sampling.

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Data in the 1970s and 1980s from Lower Parker Horn (7) and South Lake (9) are presented as averages of the twice-monthly samples during May–September and were used to indicate lake conditions. Those areas were most affected by CRW that began in April, although CRW reached into Cascade (8) and RFA (12). During April–September, lower CC (4) represented CRW inflow to Parker Horn mixed with normal flow containing background levels of TP in upper CC (3). Data for nitrogen and phosphorus from USBR between the mid-1990s and 2017 were from four samples collected near the outlet at South Lake during March-September. The four TP concentrations from USBR and nine (time-weighted) from MLIRD in 2017, each averaged 25 µg/L.

Specific conductance (SC) was used to trace CRW in the lake and determine % lake water or % CRW according to (Welch and Patmont, 1980; Cooke et al., 2005):

$$100\ [(LW - ELCW) / (CCW - ELCW)] = \%\ LW$$

$$\%\ CRW = 100 - \%\ LW$$

Water samples for algae identification and enumeration in 2017 were collected from the same water sampled for other constituents and preserved with Lugol's solution. Samples for algae were collected in mid-July, early September and early October. Abundance of algal taxa was determined as cells/ml and expressed as biovolume in mm^3/L based on measured cell volumes of individual species observed (Matthews et al., 2018).

Phosphorus Loading

External loading during May-September, 2017, was estimated as average surface inflow volumes from respective sources multiplied by their average TP concentrations during the five months. Earlier loading estimates during the 1970s-1980s were on a monthly basis, which was not possible in 2017 because monthly TP data on inflow streams were incomplete. Average flow and TP concentrations in CC and Rocky Ford Creek (RFC) were 1.5 and 1.71 m^3/sec and 42 and 149 µg/L TP, respectively. Total P concentrations in CC in 2017 were from the four observations by USBR, as well as by MLIRD. Average TP in CRW was 7 µg/L, observed by MLIRD, and 87 µg/L in RCW base flow was from Carroll (2006). Loading from ground water was included as 25% of total observed in 2001 (Pitz, 2003; Carroll, 2006).

Internal loading was estimated as the difference between observed whole-lake, volume-weighted (v-w) TP concentration and predicted lake TP determined by (Brett and Benjamin, 2008):

$$TP_{lake} = TP_{inflow} / (1 + T^{0.5})$$

That result was compared with estimates using the same approach with 1984-1988 loading when internal loading was determined directly by mass balance (Jones and Welch, 1990).

Results

Pattern of dilution with CRW

The average inflow rate of CRW was 325 x 10^6 m^3 over an average of 208 days during 2002-2016 (18 m^3/sec). That represents nearly 4 half-lake volumes contained in the most affected lake sections, Cascade (8), Lower Parker Horn (7), South Lake (9) and Lower Pelican Horn (10). That flow rate of CRW would replace that volume at about 2.0 %/day. Normal CC and RFC inflow of 3.7 m^3/sec together would replace that half-lake volume at less than two tenths that rate (0.4 %/day). Thus, once CRW were added during the spring and early summer, the resulting mixture of CRW and lake water would have been replaced very slowly given the relatively slow inflow rate of the two creeks – theoretically on the order of 600 days to replace 90% of the half-lake volume. Thus, some fraction of CRW remained in the lake until the following spring. Of course, ground water with relatively high SRP also enters the lake, although that flow is less than one-tenth surface inflows.

Nutrient contents in the two major surface inflows to the lake, CC and RFC, and CRW, are shown in Table 1. Both TP and SRP concentrations at the Highway 17 gauging site on CC (3) have averaged much higher over the past 14 years than in the 1980s. Nevertheless, that effect was minimized when large volumes of CRW, with low TP (19 µg/L) and SRP (4 µg/L), were transported through Parker Horn during the spring-summer periods. Inflow from RFC to upper RFA contained very high TP concentrations with a high fraction that was available SRP (79%) (Table 1). TP and SRP have remained rather constant in RFC, which is largely spring fed, but NO_3-N has doubled. Nitrate in CC also increased, but only about 40%. Flow-weighted average NO_3-N content in the two streams together increased 77% from 677 to 1196 µg/L, while TP increased only 13% from 109 to 123 µg/L.

Columbia river water is low in SC in contrast to CC (Table 2). Moses Lake is a hard-water lake, so SC is naturally high. CRW can be traced in the lake because it was around one-fourth the levels in CC, RFC and the lake (Table 2). Thus, lake SC was highly inversely correlated with CRW input over the past 41 years, because CRW inflow with very low SC replaced high SC lake water (Figure 2).

In 2017, SC at Lower Parker (7) and South Lake (9) averaged 288 µS/cm during May-September. That represents 58% CRW and 42% original lake water, and largely accounts for the low average

TP concentration of 25 µg/L observed in that area of the lake. Total P was also inversely related to CRW inflow over the past 40 years (Figure 3).

Despite the relatively low input of CRW (93 X 10^6 m^3) in 2017, SC in lower Parker Horn (7) and South Lake (9) increased from 267 to only 299 µS/cm during spring-summer, representing a decrease in CRW from 64% to 55%. That shows how slowly CRW is removed once in the lake. Most of the CRW (80%) entered during spring in 2017, which is usually the seasonal pattern. In contrast, SC was lower in 2016 averaging 254 µS/cm or 68% CRW through August, due to a two and half times more input of CRW (234 X 10^6 m^3).

Trophic state indicators

Total P markedly decreased in Lower Parker Horn (7) and South Lake (9) during the 1970s and 1980s due to the input of low-P CRW, which increased from 5 x 10^6 m^3/year in 1969-1970, before dilution, to an average of 130 x 10^6 m^3/year during 1977-1988, and TP decreased even further as CRW inputs more than doubled after 2002 averaging 325 x 10^6 m^3 (Table 3, Fig. 4). The average of 23 µg/L TP during 2002-2016 is based on four samples collected by USBR near the South Lake outlet (9) from mid-April to mid-October. Previous average TPs included both Lower Parker Horn (7) and South Lake (9), which together represents nearly half the lake volume most affected by CRW. Average spring-summer TPs from USBR data (n = 4) and the MLIRD (May-September, n = 9) in 2017 were each 25 µg/L, which tended to validate using the 15 years of USBR data as representative of Lower Parker Horn and South Lake trophic state.

Chl averaged 7 µg/L in Lower Parker Horn (7) and South Lake (9) during spring summer 2017. Using the ratio of chl:TP (0.35) during 2017, chl during 2002-2016 likely averaged about 6 µg/L

(Table 3). The ratio of chl:TP of 0.35 agrees with the average for world lakes at intermediate TP concentrations (Welch and Jacoby, 2004). These levels were much lower than in the 1970s-1980s, and correspond with lower TP.

Average spring-summer Secchi disck transparency in Lower Parker Horn and South Lake in the past ranged from 0.7 m during 1969-1970, before dilution began, to around 1.5 m during 1977-1988 – the goal for dilution with CRW (Table 3). Observed transparency was consistently 40% greater than predicted from a relationship between chl and transparency (Carlson 1977). Average observed transparency in Lower Parker Horn and South Lake in 2017 was only 1.4 m, which was 2/3 of that predicted from the low chl of 7 µg/L (Table 3).

The trophic state in the half volume of the lake most affected by CRW is now mid to lower mesotrophic, as indicated by TP and chl, compared to hypereutrophic before dilution and eutrophic after the start of dilution in the 1970s and 1980s (Table 3). However, transparency, at less than 2 m, still indicates eutrophy (Nürnberg, 1996).

Total P and chl were much higher in other areas in the lake in 2017, indicating a hypereutrophic state. Upper RFA had average TP and chl concentrations of 86 and 26 µg/L, respectively, which were slightly higher than average levels half way up RFA (12) in the 1970s-1980s of 70 and 22 µg/L, respectively. Although CRW reached half way up RFA (12) in the 1970s-1980s, CRW was usually around 30%, much less than in Lower Parker and South Lake. The chl:TP ratios in upper RFA in 2017 were around 0.30, about as expected.

Average summer in-lake ratios of NO_3-N:SRP and TN:TP have increased from 1.2 and 7.5, respectively, in 1969-1970 before dilution, to 5.2 and 7.3 during 1977-1988, to 9.5 and 17.4, during 2003-2017 (Table 4). That increase was partly due to decreased in-lake phosphorus from continued addition of low-nutrient CRW, despite high internal loading, which was 43% of total during 1984-1988 after wastewater diversion (Jones and Welch, 1990). Internal load was probably a substantial fraction in 2017 as well. Also, increased lake N:P ratios were partly due to increased flow-weighted nitrate content in CC and RFC.

Algae

Cyanobacteria were still abundant in the lake during 2017 being composed primarily of *Microcystis aeruginosa* (MA) in Lower Parker Horn (7) and South Lake (9), despite lower TP since 2000. The average abundance of MA at 7 and 9 during 2017 was 60,320 cells/ml and it was double that at the mid and upper RFA (120,740 cells/ml). Cyanobacteria averaged 51% of total plankton algal biovolume in 5 of the 6 samples collected at 7 and 9 during mid-July through early October. Percent cyanobacteria was only 5% in the sixth sample due to the large biovolume of a diatom. The average summer percentage of 43 in all 6 samples may have been lower had sampling been from May-September, because cyanobacteria have usually reached maximums in August and September as water temperature, water column stability and TP increased (Welch et al., 1992). *Aphanizomenon flos aquae* (AFA) usually preceded MA seasonally and was more abundant than MA before and after dilution began and may have been more abundant in the spring and early summer that was not sampled in 2017. Nevertheless, the dominance of MA may indicate that higher N:P ratios in the inflow and lake in recent years have favored that non-N fixing cyanobacterium.

Total cyanobacteria biovolume ranged from about 11 to 25 mm^3/L at Lower Parker and South Lake in 2017 averaging 14 mm^3/L during mid-July through early October. Total biovolume of all algae at those sites averaged 67 mm^3/L, which was due mostly to a large biomass of a diatom (*Suriella*) at South Lake in July that minimized the percent cyanobacteria to 5%. Biovolume of cyanobacteria was lower at those sites during 1986-1988, averaging about 7 mm^3/L, although their fraction was higher averaging 79% during the whole summer, May-September (Welch et al. 1992). While average biovolume of cyanobacteria at Lower Parker and South Lake was lower in 1986-1988 (7 mm^3/L) than in 2017 (14 mm^3/L), the reverse occurred with average chl, which was much higher in 1986-1988 (17 µg/L) than in 2017 (7 µg/L, Table 3). Again, that may be due to the shorter sampling period for algae in 2017 than in 1977-1988. Cyanobacteria, including both AFA and MA, were more abundant in the mid (12) and upper RFA in 2017, than at other sites, at biovolumes ranging from 14-144 mm^3/L, averaging 74% of total algal biomass at those two sites, where TP and chl were 58 and 15 µg/L, respectively.

Phosphorus Loading

External loading was 6,950 kg from inflow streams and CRW in 2017, including a percent of total for groundwater observed in 2001 (Pitz, 2003). Inflow volume was 168 x 10^6 m^3 including 18% from groundwater (Carroll, 2006). The resulting flow-weighted inflow TP was 41 µg/L and predicted lake TP was 20 µg/L, assuming a period water retention time of 0.92 yr. The difference between predicted lake TP and the observed v-w whole-lake TP of 51 µg/L is 31 µg/L, which represents an estimate of the contribution from internal sources. Thus, internal loading still persisted at about 60% of total for the whole lake in 2017.

The average fraction of total loading from internal with 1984-1988 data was 45%, determined as the difference between predicted and observed of 28 µg/L, using the same model to estimate lake TP from inflow TP concentration and water residence time (Brett and Benjamin, 2008). The average fraction of total loading that was internal in 1984-1988, determined directly from mass balance, was 43%, which is similar to that estimated using the model approach (45%). Internal loading may have been underestimated because whole-lake TP was calculated using 5-m samples, although the unrepresented, deeper, higher-TP water is only about 10% of total volume, which occurs in the CRW affected area.

Positive internal loading, even after over 30 years of CRW input, and after wastewater diversion, represents a continual threat to lake water quality, particularly as the summer progresses. Continual CRW input annually is essential to dilute internal as well as external loading to maintain acceptable lake water quality for recreation and water use and minimize the frequency and abundance of cyanobacteria blooms. Without CRW in 2017, average v-w whole lake TP would likely have been on the order of 90 µg/L, three to four times the 25 µg/L average in 2017 and the 23 µg/L average during 2002-2016 (Table 3). That reasoning is based on the average lake TP concentrations at South Lake in 1996-1998 of 93 µg/L when CRW inputs were relatively low averaging 77 x 10^6 m^3, which was one fourth the 2002-2016 average (Figure 4; Table 3).

Without CRW, external loading in 2017 would have been 6,300 kg and inflow TP 84 µg/L with a predicted whole lake TP of 67 µg/L. Nevertheless, TP was surprisingly low in 2017, given the relatively low CRW input of 93 x 10^6 m^3 (Table 3). There has been considerable variability in lake TP for any given CRW inflow, although it was consistently low for inputs above 250 x 10^6 m^3, as indicated in Figure 3. Variability in lake TP was substantial even with CRW inputs between 100 and 250 10^6 m^3 (Figure 3). Much of that was probably due to internal loading, which varied by ± 97% during 1977-1988 largely due to variable year-to-year water column mixing (Jones and Welch, 1990). Internal loading accounted for an average lake TP of 90 µg/L in 1985 even with a CRW input of 170 10^6 m^3, due to low water column stability (Figure 4).

Discussion

The addition of high volumes of low-nutrient CRW to central Moses Lake over the past 41 years has consistently maintained a mesotrophic state, especially with the much higher volumes over the past 18 years. CRW inputs tripled during that recent period compared to the first 10 years of the dilution project. Without CRW input, TP concentration in the Parker Horn and South Lake area in 2017 would probably have been more than triple the observed, on the order of 90 µg/L instead of 25 µg/L, judging from TP at South Lake when CRW inputs were very low in 1996-1998. Even with substantial inputs of CRW, lake TP was relatively high due usually to high internal loading.

Inputs of CRW have varied greatly over the past 41 years since the dilution program began; average input since 1977 was 228 x 10^6 m^3 ± 50%. Much of the reason for that variation is the amount of inflow from streams, especially CC. For example, the low average input of CRW in 1996-1998 was 24% of that during 2002-2016, while annual mean CC inflow in 2002-1016 was 29% of that in 1996-1998. Thus, more CRW was routed through the lake in dry years with low CC runoff, than in wet years, due largely to irrigation demand and downstream storage capacity. Nevertheless, the relatively long retention time of CRW in the lake, presumably lasting from one year to the next, has allowed rather stable spring-summer TP concentrations even if CRW inputs were low, such as 2017 with 25 µg/L and only 93 x 10^6 m^3 of CRW. To maintain a mesotrophic state in half the lake most affected by dilution, CRW input should be at least 200 10^6 m^3. Inputs above that resulted in about 2/3 of the spring-summer TP concentrations being less 30 µg/L or less (Fig. 3).

Despite the lowered trophic state, cyanobacteria still dominated the spring-summer algal crop during 1977-1988, although less (65%) than before the dilution project began (98%, Welch et al., 1992; Welch, 2009). Cyanobacteria were about half of total algal biovolume in 2017 when the lake was mesotrophic. However, their fraction may have been lower if data were available from spring-early summer, before temperature, water column stability and TP increased; TP was 23 µg/L during May-August and 40 µg/L in September. Cyanobacteria biovolumes and percent composition (74%) were much higher in mid to upper RFA in 2017, while TP and chl were 58 and 15 µg/L, respectively. Transport of CRW into RFA was much less effective; middle RFA (12) contained only about 30% CRW during spring-summer in 1977-1988.

That cyanobacteria averaged 8 mm^3/L and 65% of algal biovolume at Lower Parker and South Lake during 1977-1988 was not surprising, because TP and chl averaged 65 and 15 µg/L, respectively (Welch et al., 1992). The risk for cyanobacteria dominance was observed to increase sharply at TP above 30 µg/L (Downing et al., 2001). Thus, cyanobacteria at about half total algal biovolume and 14 mm^3/L in 2017 is surprising, given TP at only 25 µg/L, but may be due to no spring, early summer algal data when TP was relatively low.

Inflow NO_3-N concentrations in the two main inflow streams was higher in recent years. While CRW was also low in NO_3-N, as well as TP and SRP, ratios of NO_3-N:SRP and TN:TP in stream inflows (CC and RFC) during 1986-1988 and 2003-2017 were at or above the Redfield ratio of 7.2. Thus, P was still the key inflow nutrient determining trophic state, although higher nitrate in the inflows may have affected cyanobacteria species composition.

In-lake NO_3-N:SRP ratios were usually well below the Redfield ratio in the 1970s-1980s, indicating short-term growth rate limitation by N. Nitrate usually decreased to below detection during summer and that was still the case in 2017. However, the in-lake TN:TP ratios in the 1970s-1980s were maintained at or slightly above the Redfield ratio despite the low in-lake NO_3-N:SRP ratios and substantial inputs of P internally from ground water and undiluted RFC. The added N to maintain TN:TP at or above the Redfield ratio was likely produced by N-fixing AFA, which amounted to 90% of total summer TN loading during 1981-1982 (Welch, 2009). That N-fixer was usually the dominant cyanobacteria before dilution and during the 1970s-1980s; e.g., it was 48% of average total algal biomass during early summer and 94% on July 8 in 1986 while MA was nearly absent (Bouchard, 1989; Welch et al., 1992). AFA was present in July 2017, but at a low fraction of total algal biovolume and not present in September. Dominance by MA and low biovolume of AFA in Moses Lake in 2017 may have been due to much higher N:P ratios in the inflow and lake in recent years.

Increased P input usually favors N-fixing cyanobacteria (Havens, 1995; Gophen et al., 1999). Also, there is agreement among most investigators that N fixers have the capacity to supply the N needed to meet available P, if N input were low or if N, as well as P, input were decreased (Patterson et al., 2011; Beversdorf et al., 2013). Long-term, continuous P-only loading to lake

227, Ontario, has kept the lake eutrophic, with N supplied by N-fixing cyanobacteria (Schindler et al., 2012, 2016; Higgins et al., 2017). Similarly, N-fixation by AFA supplied nearly half the N input to hypereutrophic Clear Lake, California (Horne and Goldman, 1972). Also, mesocosm experiments in an eutrophic lake demonstrated the capacity of N fixation to supply enough N to match the growth potential from P (Vrede et al., 2009). Apparently, N-fixers have that capacity even if TN:TP ratios are above the Redfield ratio (Tonno and Nõges, 2003). Non-N-fixing MA often succeeds AFA during summer, taking advantage of fixed N (Baversdorf et al., 2013). Throughout nearly 50 years of trophic state change from hypereutrophy to mesotrophy, the in-lake TN:TP ratio has remained at or above the Redfiled ratio with P controlling algal biomass, even though N was growth rate limiting, with the added N coming from N fixation.

Conclusions

The 41-year record of low-P CRW input to Moses lake has demonstrated the effectiveness of lowering lake TP and trophic state when CRW volumes were adequate. If not adequate, lake TP was less effectively lowered, even if inflow TP were decreased proportionately and % CRW were substantially and predictably increased, due to TP from internal loading and ground water. Inputs of at least 200 x 10^6 m^3 usually produced lake water quality near the mesotrophic state. Nevertheless, cyanobacteria have continued to dominate, although their % of total biovolume declined. Their persistence is surprising given the relatively low TP, as well as the increased soluble and total N:P ratios in the inflow and lake.

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Table 1. Average nutrient concentrations in inflows during spring-summer in 1986-1988 (Welch et al. 1992) compared with 2003-2017 in ().

	TP	SRP	NO_3-N
Crab Creek	47 (75)	7 (40)	685 (950)
Rocky Ford Creek	164 (165)	107 (131)	669 (1405)
East Low Canal (CRW)	19 (7[1])	4 (NA[1])	30 (10[1*])

*detection limit; [1]2017 only

Table 2. Average specific conductance in µS/cm used as tracer of CRW (Columbia River water) throughout Moses Lake.

CRW	142	
Moses L (pre dilution)	445	(1969-1970)
Crab Creek	491	
Rocky Ford Cr	371	

Table 3. Spring-summer Columbia River water (CRW) inflows, half-lake volumes (77.8 X 10^6 m^3) replaced and average spring-summer TP and chl concentrations and transparency (SD) in South Lake (9) and Lower Parker Horn (7). USBR data from 9 only. *Chl estimated from chl:TP = 0.35. **Spring-summer average TP from 9 (USBR) equaled TP at 7 and 9 together (MLIRD) in 2017.

		10^6 m^3	AFX10^3	0.5 Lake Volumes	µg/L		m
	Years	CRW	CRW	0.5	TP	CHL	SD
UW	69-70	5	4	0.37	152	58	0.7
UW	77-84	105	85	1.36	74	21	1.4
UW	86-88	178	144	2.3	41	17	1.6
DOE	01	284	230	3.7	19	11	NA
USBR	02-16	325	263	4.8	23	6*	NA
MLIRD	2017	93	76	1.2	25**	7	1.4

Table 4. Average ratios of soluble and total N and P during spring-summer before and after dilution began in 1977 (from Welch 2009) compared with recent years (data from USBR) and the Redfield ratio of 7.2 by weight.

	1969-1970	1977-1988*	2003-2017
NO_3-N:SRP	1.2	5.2	9.5
TN:TP	7.5	7.3	17.4

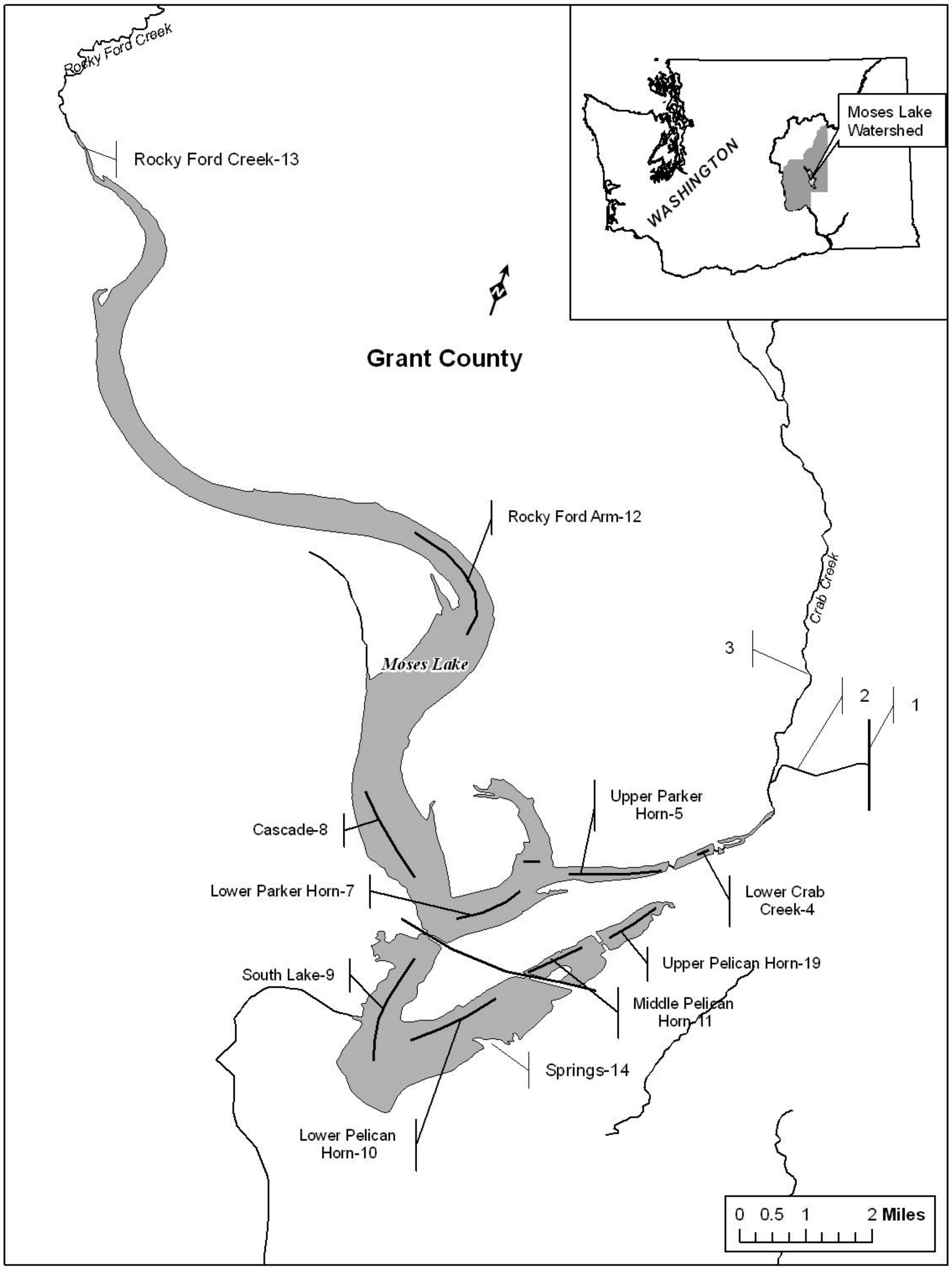
Figure Titles

Figure 1. Sampling sites during 2017, most similar to those during 1969-1970 and 1977-1988 (Welch et al., 1989). Site 1 is East Low Canal and site 2 is Rocky Coulee Wasteway, routs for CRW inflow.

Figure 2. Relation between CRW input and spring-summer average specific conductance in µS/cm in Lower Parker Horn (7) and the South Lake (9) during 1969-1988 from Welch et al. (1992) and South Lake near north outlet during 1995-2017 from USBR.

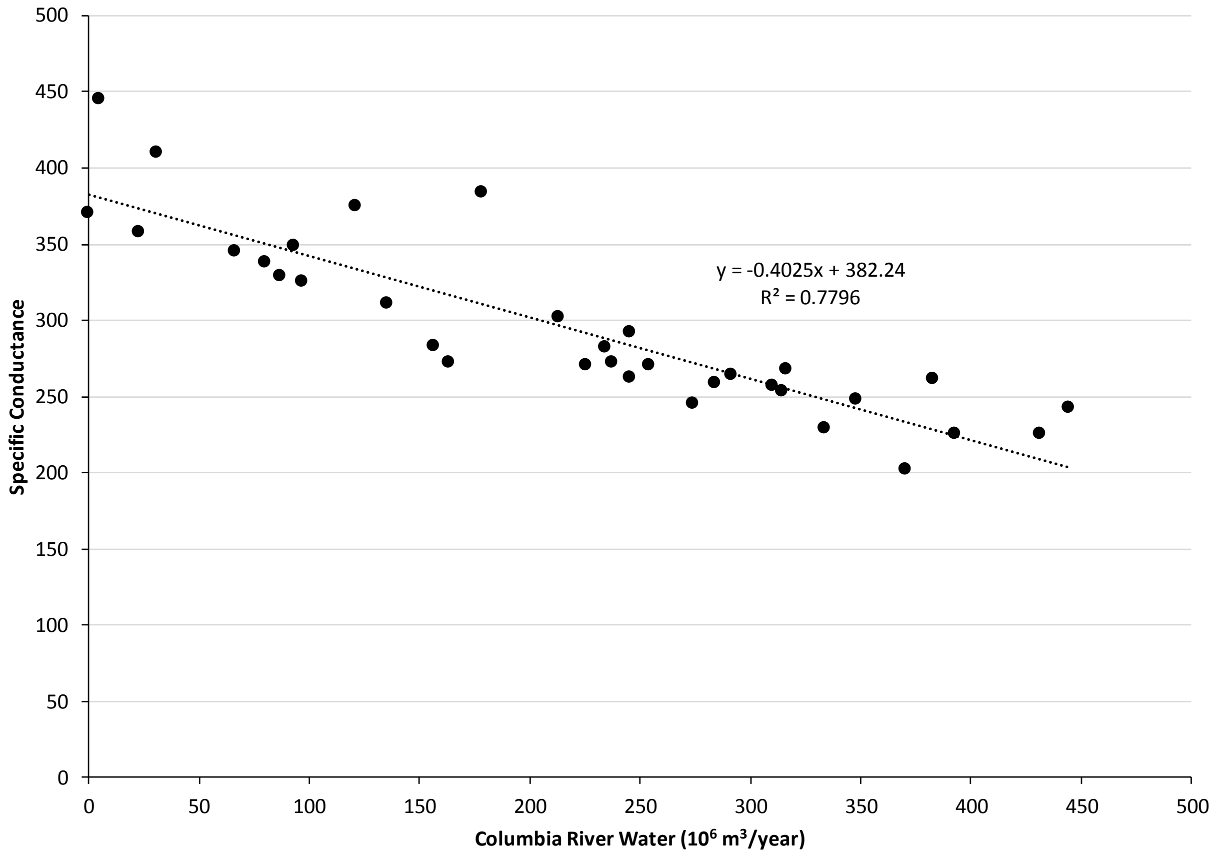
Figure 3. Relation between CRW input and spring-summer average TP concentration in Lower Parker Horn (7) and the South Lake (9) during 1969-1988 from Welch et al. (1992) and South Lake near north outlet during 1995-2017 from USBR.

Figure 4. Inflow of Columbia River water into Moses Lake over nearly 50 years and spring-summer average TP concentration in Lower Parker Horn (7) and the South Lake (9) during 1969-1988 (May-September) from Welch et al. (1992) and South Lake near north outlet during mid-April to mid-October, 1995-2017 from USBR.



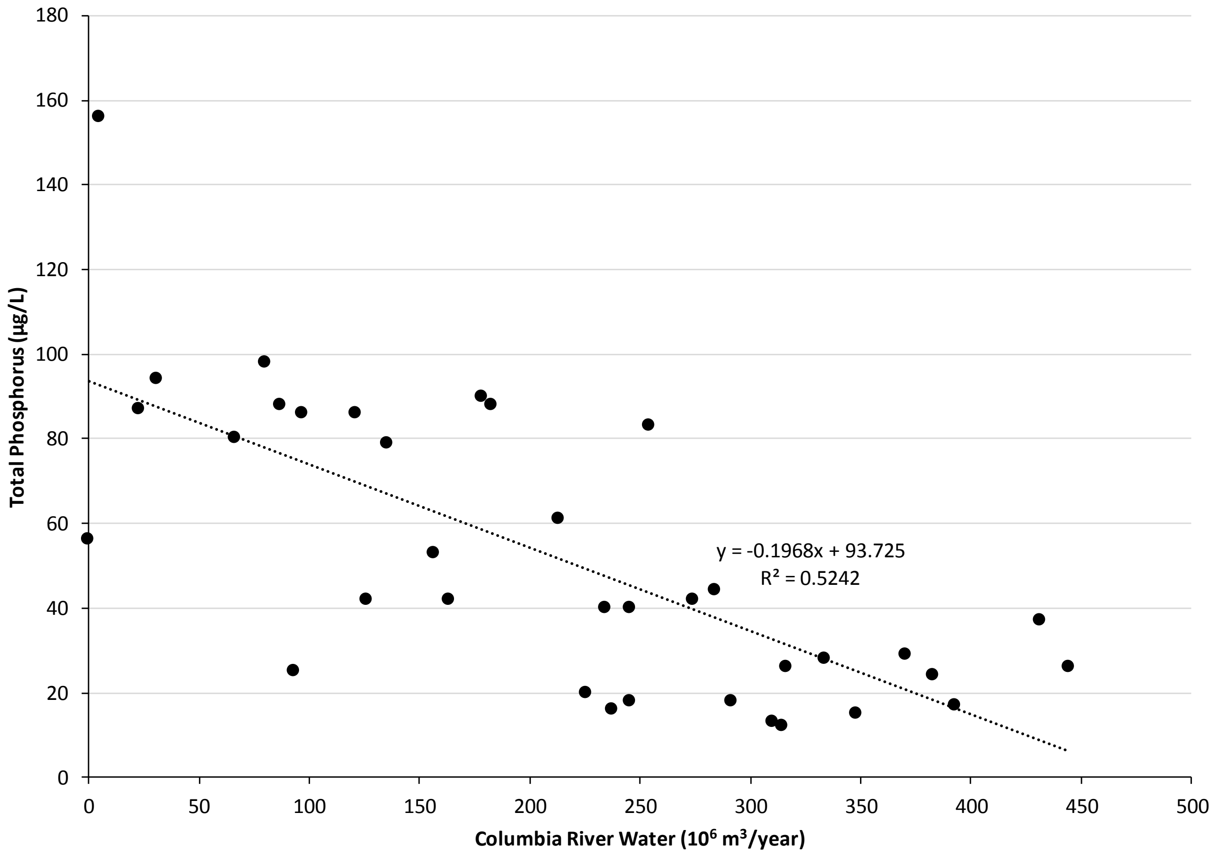
Rocky Ford Creek
Rocky Ford Creek-13
Moses Lake
Watershed
WASHINGTON
Grant County
Rocky Ford Arm-12
Crab Creek
Moses Lake
3
2
1
Upper Parker
Horn-5
Cascade-8
Lower Parker Horn-7
Lower Crab
Creek-4
Upper Pelican Horn-19
South Lake-9
Middle Pelican
Horn-11
Springs-14
Lower Pelican
Horn-10
0 0.5 1 2 Miles

Figure 1



500
450
400
350
300
250
200
150
100
50
0
Specific Conductance
y = -0.4025x + 382.24
R² = 0.7796
0
50
100
150
200
250
300
350
400
450
500
Columbia River Water (10⁶ m³/year)

Figure 2



180
160
140
120
100
80
60
40
20
0
Total Phosphorus (µg/L)
0
50
100
150
200
250
300
350
400
450
500
Columbia River Water (10⁶ m³/year)
y = -0.1968x + 93.725
R² = 0.5242

Figure 3

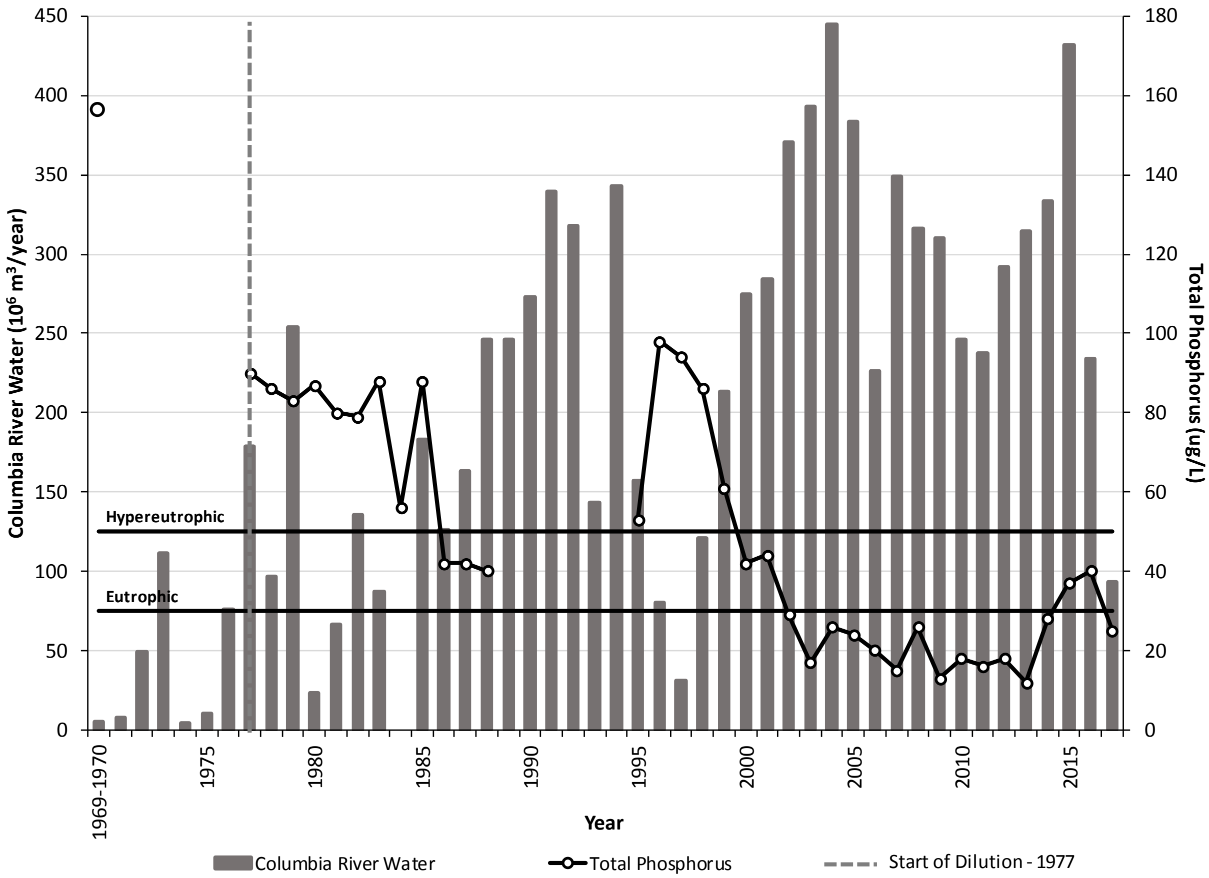

Columbia River Water (10^6 m^3/year)
Total Phosphorus (ug/L)
Hypereutrophic
Eutrophic
Year
Columbia River Water
Total Phosphorus
Start of Dilution - 1977
450
400
350
300
250
200
150
100
50
0
180
160
140
120
100
80
60
40
20
0
1969-1970
1975
1980
1985
1990
1995
2000
2005
2010
2015

Figure 4